

Modeling potential erosion due to the Cerro Grande Fire with a GIS-based implementation of the Revised Universal Soil Loss Equation

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Abstract. Erosional processes directly influenced by wildland fire include reduction or elimination of above-ground biomass, reduction of soil organic matter, and hydrophobicity. High fuel loads promoted by decades of fire suppression in the U.S. increase the duration and intensity of burning, amplifying these effects. The Cerro Grande fire (6–31 May 2000) consumed approximately 15 000 hectares around and within the town of Los Alamos, New Mexico, USA. Private and public infrastructure including Los Alamos National Laboratory are at continuing risk due to increased threats of upstream erosion. We use a geographic information system (GIS) based implementation of the Revised Universal Soil Loss Equation (RUSLE) to model pre- and post-fire soil loss conditions and aid erosion risk analysis. Pre- and post-fire vegetation cover data layers were generated from Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM) data. Based upon annual average rainfall amounts we estimate that subwatershed average pre-fire erosion rates range from 0.45 to 9.22 tonnes ha^{−1} yr^{−1} while post-fire erosion rates before watershed treatments range from 1.72 to 113.26 tonnes ha^{−1} yr^{−1}. Rates are approximately 3.7 times larger for 50 year return interval rainfall amounts. It is estimated that watershed treatments including reseeding will decrease soil loss between 0.34 and 25.98% in the first year on treated subwatersheds. Immediately after the fire an interagency Burned Area Emergency Rehabilitation (BAER) team produced initial estimates of soil erosion. Our estimates of average erosion rates by subwatershed were in general larger than those initial estimates.

Additional keywords: wildland fire; watershed; cover factor; rainfall–runoff erosivity factor; soil erodibility; slope length and steepness factor.

Introduction

Erosion due to effects of wildfire is of major concern to land managers. Soil losses following wildfire are influenced by the amount of vegetative cover removed, soil type, topography, and intensity of storms (Cannon and Reneau 2000). Of these factors, removal of vegetative cover and slope are the most important under intense rainfall (Prosser and Williams 1998). In some vegetation and soil types, secondary compounds in the surface litter are volatilized under intense fire conditions, creating a hydrophobic layer in the soil, increasing runoff and erosion (DeBano 1981; Robichaud 2000). Erosion rates are also generally correlated with fire severity, but site-specific factors may enhance or decrease rates (Robichaud and Waldrop 1994; Cannon and Reneau 2000). Large increases in erosion rates of several orders of magnitude after wildfire have been observed (Morris and Moses 1987). Flooding, debris flows, and severe channel incising can result from post-fire rainfall events. Such large effects are

dependent on storm intensity, which in the Southwest U.S. can be highly localized and variable (Wohl and Pearthree 1991; Cannon and Reneau 2000).

Our objective in this paper is to demonstrate a method of deriving estimates of soil erosion that can be used in post-fire erosion risk analysis through the use of established modeling methods, geographical information systems (GIS), and the best available data. One of the most widely used erosion prediction models is the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1965, 1978) and the updated version, the Revised Universal Soil Loss Equation (RUSLE) (Renard *et al.* 1997). The original USLE was empirically derived from over 10 000 plot-years of data. Both USLE and RUSLE compute erosion as a linear combination of factors:

$$A = R \times K \times L \times S \times C \times P, \quad (1)$$

where A is the computed spatial and temporal average soil loss per unit area, expressed in the units selected for K and for

the period selected for R ; R is the rainfall-runoff erosivity factor; K is the soil erodibility factor; L is the slope length factor; S is the slope steepness factor; C is the cover-management factor; and P is the support practice factor (Wischmeier and Smith 1965, 1978). Applications for which the equation was specifically designed and field tested include:

- (1) Predicting average annual soil movement for a given field slope;
- (2) Guiding selection of conservation practices;
- (3) Estimating reduction in soil loss attained from conservation practices; and
- (4) Estimating soil losses from rangeland, woodland, and recreational areas.

The USLE was designed to predict soil loss due to sheet and rill erosion on straight hill slopes. The USLE equation can, however, be used properly at watershed and broader scales by dividing complex watersheds into smaller areas over which the USLE factors can be defined (Wischmeier 1976). Procedures have been developed to calculate the soil loss on complex slopes by dividing irregular slopes into uniform segments (Foster and Wischmeier 1974). Deposition and channel erosion are not modeled and therefore the USLE does not model sediment yields at catchment and broader scales; rather, the model approximates the amount of soil moved (Wischmeier 1976).

Infiltration rates of undisturbed forests are generally high. After fire, infiltration rates decrease 2.5 to 7 times in severely burned areas compared with unburned areas (Campbell *et al.* 1977; Martin and Moody 2001). Reduced infiltration rates promote increased overland flow and higher sediment transport capacity (Dunne and Leopold 1978). Reduced vegetation cover, controlled by fire severity, leads to increased splash erosion and runoff (Zwolinski 1971; Dunne and Leopold 1978). Increased runoff and erosion rates can lead to flooding and/or debris flows, determined predominantly by fire severity, watershed topography, and storm intensity (Campbell *et al.* 1977; Diaz-Fierros *et al.* 1987; Cannon and Reneau 2000). Of these variables fire severity, which influences the cover factor, varies the most between pre- and post-fire conditions. All other factors can be presumed to be constant. Therefore, although USLE only predicts sheet and rill erosion, estimated rates can suggest whether a watershed may be at risk to post-fire flooding and/or debris flows.

Many researchers have demonstrated GIS versions of USLE type models at catchment and larger scales since the 1980s (Diaz-Fierros *et al.* 1987; Ventura *et al.* 1988; Millward and Mersey 1999; Yitayew *et al.* 1999). Two advantages of using GIS in modeling erosion is that it provides the framework for deriving the USLE factors, and can apply data at fine resolutions over broad scales (Wilson and Lorang 2000). GIS has been used most commonly to derive the cover (C) and topographic factors (L and S).

Although several researchers have implemented GIS-based USLE type models, we do not believe there is any published work implementing RUSLE with hydrophobic soils and post-fire watershed treatments in forested mountainous terrain for post-fire erosion risk analysis. We derive spatial variables for all RUSLE factors from satellite data and existing databases. We also integrate pre- and post-fire field level plot data with satellite derived vegetation and canopy consumption maps to produce spatially varying pre- and post-fire cover factors in a forested landscape.

There are implications to applying RUSLE based methods to a mountainous landscape-level application; RUSLE predicts only sheet and rill erosion, ignoring channel erosion and deposition processes (Renard *et al.* 1991). Numerous other researchers are working on improving landscape level erosion models. We do not attempt here to improve on established erosion modeling methods. Although actual erosion rates may differ from those presented here, predicted rates can make a useful contribution to risk analysis.

Data and methods

Study area

The Cerro Grande fire occurred on the eastern slopes of the Jemez Mountains, approximately 40 km north-west of Santa Fe, New Mexico. The fire encompassed the western portions of Los Alamos County on the east, including portions of Los Alamos National Laboratory (LANL), northern portions of Bandelier National Monument (BNM) on the south, the Santa Fe National Forest to the crest of the Sierra de los Valles on the west bordered by private land, and the Santa Clara Indian Reservation on the north. Elevations in the study area range from about 2000 m in pinyon-juniper woodland to over 3100 m in mixed conifer forest. Precipitation levels range from about 46 cm per year at the Los Alamos town site to 76 cm per year at the highest elevations in the Sierra de los Valles, with 40% occurring during monsoon thunderstorms in July and August (Bowen 1990).

The Sierra de los Valles is a semicircular mountain chain composing the eastern remnant of an ancient volcanic caldera and forming the western edge of the study area. The volcanic highlands of the Sierra resulted from Tertiary volcanic flows and eruptions. Two major eruptions occurring between 1.1 and 1.4 million years ago destroyed the entire volcano, leaving the current rim of the caldera. Much of the ejected pumice and rhyolite ash was deposited immediately east of the volcano, forming the Bandelier tuff and the Pajarito Plateau on the eastern side of the study area. Erosion has subsequently dissected the Pajarito Plateau into mesas, separated by steep canyons (Kelly 1978).

About 80% of the soils in the study area are Alfisols, Entisols, or Inceptisols, with Alfisols occurring at the higher elevations. The dominant soils in the study area are moderately deep to shallow on steeper slopes and deep on gentle

slopes. Surface textures range from fine sandy loam to gravely fine sandy loam, with coarse surface fragments ranging between 5 and 40% gravels, cobbles or stones. Some soils on older surfaces or at high elevations have clay loam subsoils. Erosion hazard is high to very high when vegetation cover is removed from these soils (Nyhan *et al.* 1978; BAER 2000).

Pinyon–juniper woodlands are the dominant vegetation community at the lower elevations of the study area, between 1700 m and 2100 m. Primary tree species are one-leaf pinyon (*Pinus edulis*) and one-seed juniper (*Juniperus monosperma*). Ponderosa pine (*Pinus ponderosa*) forests extend from 1900 m in protected canyons to 2400 m on the lower slopes of the Sierra de los Valles. Ponderosa pine is the dominant tree species in this elevation range, with pinyon and juniper present at lower elevations. At higher elevations, Douglas-fir (*Pseudotsuga menziesii*) can be found intermixed in the overstory. Mixed conifer forest intergrades with ponderosa pine communities above 2100 m, and extends to the top of the Sierra de los Valles. Douglas-fir, white fir (*Abies concolor*), and limber pine (*Pinus flexilis*) are the dominant trees at the lower mixed conifer range with Englemann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) occurring at the higher elevations. Aspen (*Populus tremuloides*) occurs at the same elevations, intermingled with mixed conifer forest, or may be dominant in previously burned areas (Foxx and Tierney 1980; Balice 1998).

Model elements

Rainfall-runoff erosivity factor

The rainfall-runoff erosivity factor (R) is a significant factor in modeling erosion. When all other RUSLE factors are held constant, soil losses are directly proportional to R (Diaz-Fierros *et al.* 1987; Renard *et al.* 1997). Many previous studies have used existing erosivity maps to determine R , usually a constant over the study area (Wilson and Lorang 2000). Average annual precipitation for our study site is about 46 cm per year at the lower elevations and almost doubles to 76 cm per year at the highest elevations. We thus sought a spatial representation of R . Existing erosivity maps were not of sufficiently fine scale to provide the spatial detail to meet our requirements. Rain gauge data could not be used to compute a spatial erosivity surface. There are only two permanent weather stations with historical data within the study area and they occur at the lower elevations. Wischmeier (1974) used data from the Western US Plains States to develop an equation to estimate the erosivity factor:

$$R = 27.38 \times P^{2.17}, \quad (2)$$

where P is the maximum 6 h rainfall amount expected to occur within a 2 year time span. Precipitation values are available for the two weather stations for 2, 5, 10, 25, 50 and 100 year return intervals and 15 min, 30 min, 1 h, 3 h, 12 h, daily and annual time periods (Bowen 1990). With these data we

developed a regression equation of annual rainfall to 2 year 6 hour rainfall amounts ($R^2 = 0.996$). To develop an erosivity surface for the study area we obtained modeled annual average precipitation data from the Parameter-elevation Regressions on Independent Slopes Model (PRISM) (Daly *et al.* 1994). We interpolated these data from the original 4 km resolution to 1 km resolution, applied our regression equation and then equation (2) to obtain an erosivity surface (Fig. 1a). For risk analysis, we computed an erosivity surface for the 50 year return interval rainfall amounts. The annual average rainfall surface was scaled up by a factor of 1.76, the ratio of the 50 year annual rainfall recorded at the weather stations to the 2 year annual rainfall. The 50 year erosivity surface was then calculated using the same procedures as the average annual rainfall erosivity surface.

Soil erodibility factor

Soil erodibility as defined in USLE and RUSLE is due to a combination of splash during rainfall, runoff, and infiltration (Renard *et al.* 1997). Since the USLE is a common soil conservation tool, most soil surveys include the soil erodibility factor K . The Soil Survey Geographic (SSURGO) Database is the only survey that covers the entire study area, though at a very coarse scale (1:250 000). Portions of the study area are under management of six different agencies, most of which have electronic digital copies of portions of soil surveys at much finer scales. A soil erodibility surface was generated by combining portions of six different soil databases (Fig. 1b): (1) USDA Forest Service Terrestrial Ecosystem Survey of the Santa Fe National Forest (Miller *et al.* 1993); (2) Los Alamos National Laboratory (Nyhan *et al.* 1978); (3) Bandelier National Monument (EEC 1974, 1978; SCS 1975); (4) Rio Arriba County (NRCS 1978); (5) State Soil Geographic Database (STATSGO) (SCS 1994); and (6) Soil Survey Geographic Database (SSURGO) (NRCS 1995).

The SSURGO data at a scale of 1:24 000 are currently the best soils data for the Valles Calderas National Preserve immediately to the west of the Cerro Grande Fire, but portions are within the study area. The soil survey for Rio Arriba County, which includes the Santa Clara Indian Reservation, has never been published. The Burned Area Emergency Rehabilitation (BAER) team digitized the portion of the Reservation within Cerro Grande Fire perimeter. The STATSGO data at 1:250 000 were the best data for the un-digitized portion of the Santa Clara and San Ildefonso Reservations on the east side of the study area. All other soils data were at 1:24 000 or finer scales. Although the erodibility surface was generated from soil surveys at various scales, the portions of the study area within the Cerro Grande Fire perimeter were at the finest scales available. Soil K factors which were not included in the above soil surveys were estimated by local soil scientists for the BAER team and were also used in this study.

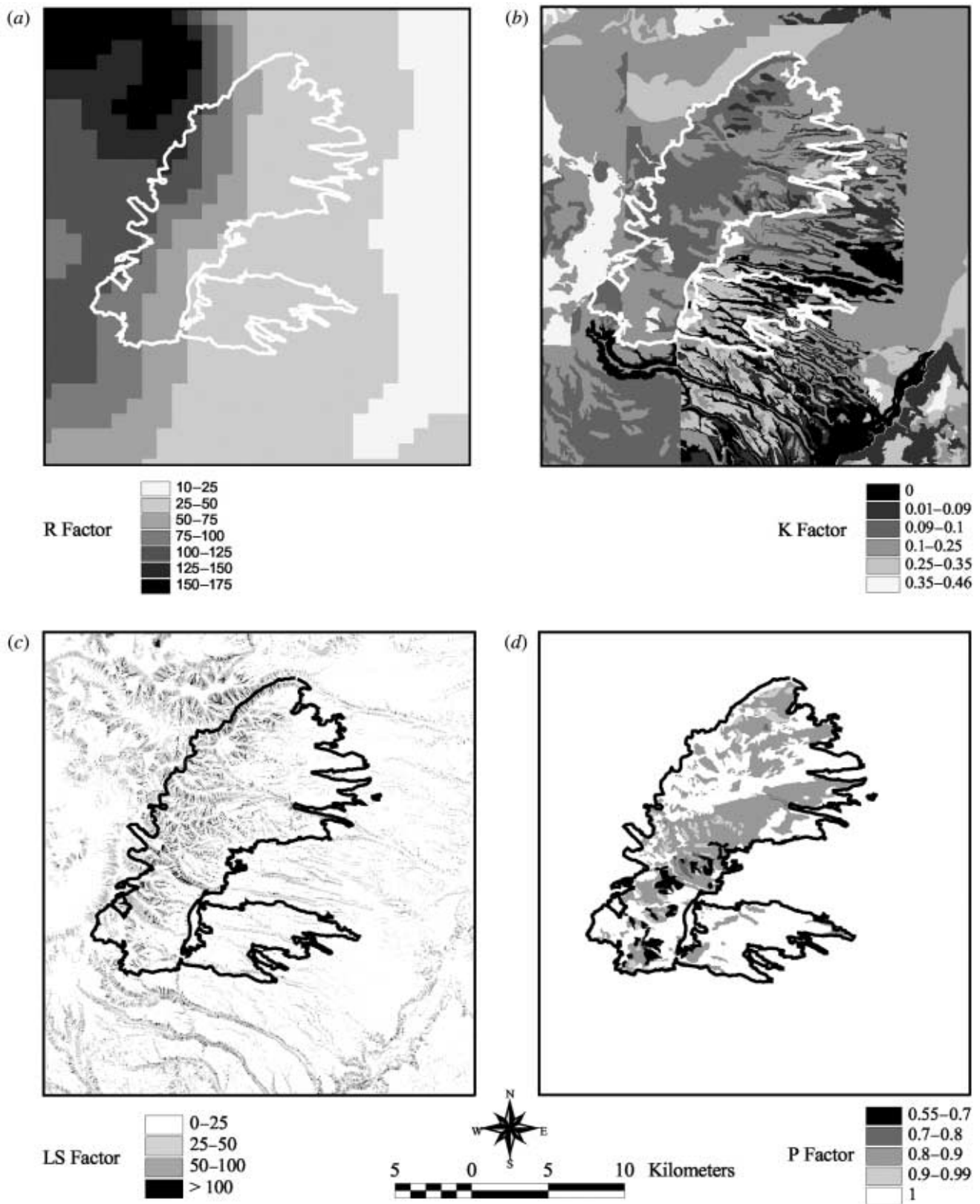


Fig. 1. RUSLE model factors: (a) erosivity (R), $\text{MJ mm ha}^{-1} \text{h}^{-1} \text{yr}^{-1}$; (b) soil erodibility (K), $\text{t ha h ha}^{-1} \text{MJ}^{-1} \text{mm}^{-1}$; (c) combined slope length and slope with rill equal to inter-rill for severely burned areas (LS), unitless; (d) post-fire watershed treatments and seeding (P), unitless.

Hydrophobic soils

Fire-induced hydrophobic soils have been a concern of watershed managers since first identified in the 1960s. Debate persists over the impact of fire-induced hydrophobicity on runoff and erosion (DeBano 2000). Some researchers have seen little erosion effects at the watershed scale (Prosser and Williams 1998). Others have reported decreases in soil infiltration rates by factors of up to 7 (Martin and Moody 2001). While hydrophobic soils may increase erosion rates, reductions in the cover factor probably dominate the increase in erosion rates seen after wildfire. Undisturbed forested conditions may have cover factors as low as 0.001 while devegetated forests may have cover factors as high as 0.45 (Wischmeier 1975).

The BAER team observed that, in general for areas with high burn severity, hydrophobic soils occurred on 75–90% ponderosa pine sites with north aspects, 25–35% ponderosa pine sites with south aspects, 25–35% mixed conifer sites with north aspects, and 5–15% mixed conifer sites with south aspects (BAER 2000). Although the existence of hydrophobic soils after wildfire has been well documented, we could not find any research previously published on how to include this condition in the USLE or RUSLE equations. Wischmeier and Smith (1978) published a nomograph for determining the soil erodibility factor as a function of percentage sand, soil structure, organic matter, and soil permeability. As permeability decreases from rapid (>6 cm/h) to very slow (<0.1 cm/h), the most severe case allowed in the nomograph, the K factor increases by approximately 0.12 (Renard *et al.* 1997). Infiltration rates were seen to decrease by no more than a factor of 7 from pre- to post-fire conditions within the Cerro Grande Fire (Martin and Moody 2001). Since the objective of this paper is to produce estimates of soil erosion for risk analysis, we chose to be conservative in our calculations and add 0.12 to the K factor for all ponderosa pine sites with high burn severity, since those were predominately the areas exhibiting hydrophobic conditions. In our study area the average K factor for all ponderosa pine sites with high burn severity was 0.143. Therefore by adding 0.12 to K , the soil erodability factor on average almost doubled.

Slope length and steepness factors

One of the potentially most important improvements in GIS-based USLE models over non-GIS models is the computation of the LS factor (Wilson and Lorang 2000). In manual calculation of USLE, slope length (L) is usually computed as a straight line. However, in a two-dimensional situation, slope length should be replaced with the upslope drainage area per unit of contour length, i.e. the upslope contributing area (Moore and Wilson 1992; Desmet and Govers 1996a). GIS technology allows for the calculation of the upslope contributing area through the use of digital elevation

models (DEM), so that complex topography may fully be accounted for.

High resolution DEM data are required for spatially distributed hydrologic modeling in complex terrain (Mitasova *et al.* 1996; Gao 1997). Larger grid sizes of 30–90 m tend to underestimate the slope curvature. DEMs of 10 m horizontal resolution have been found to provide a substantial improvement over 30 and 90 m data; however, 2 or 4 m data provide only marginal improvement even in moderate to steep topography (Zhang and Montgomery 1994). Our study area is rather large and 10 m DEMs presented an acceptable compromise of accuracy vs. storage space and computational time. We acquired from the U.S. Geological Survey 1:24 000 Level 2 DEMs at 10 m resolution to generate the topographic LS factor. All subsequent modeling was performed at 10 m resolution.

Algorithms proposed to calculate the upslope contributing area can be divided into two groups: single flow direction, and multi-flow direction. Single flow direction algorithms transfer all runoff/eroded material to a single cell downslope, thereby enabling only parallel and convergent flow patterns. Multiple flow algorithms divide the flow out of a cell over several receiving cells, allowing for divergent flow. Multiple flow algorithms have been shown to produce more spatially accurate estimates of the upslope contributing area than single flow algorithms, resulting in larger average upslope contributing values per watershed (Desmet and Govers 1996b; Wilson and Lorang 2000). We chose to use a computer program readily available over the Internet to calculate the upslope contributing area and LS factor (Desmet and Govers 1996a, 1996c). Advantages of this program include: (1) it is easy to use; (2) it implements a well known multiple flow algorithm (Quinn *et al.* 1991); and (3) it implements the RUSLE L and S equations (McCool *et al.* 1987, 1989). The RUSLE L equation allows for varying the rill erosion (surface flow) to inter-rill erosion (raindrop impact) ratio. A low rill to inter-rill ratio represents conditions such as rangeland and undisturbed forest where inter-rill erosion is greater than rill erosion and infiltration is high (McCool *et al.* 1989; Renard *et al.* 1997). We calculated an LS factor with a low rill to inter-rill ratio of 0.5 for modeling pre-fire erosion and erosion immediately after the fire before rills are able to form. We calculated another LS factor with a rill to inter-rill ratio of 1.0 for severely burned areas and 0.5 for all other burn severities to model erosion conditions when rills had formed due to post-fire rains (Fig. 1c).

Post-fire watershed treatments

In the weeks after the fire, the BAER team and others applied treatments to high and moderately burned portions of the watershed to reduce erosion and surface runoff. Planned locations areas for application of each treatment were mapped by the BAER team in a GIS. Watershed specialists on the BAER team estimated how much each treatment would

Table 1. Post-fire watershed treatment support practice (*P*) factors

| Treatment | <i>P</i> factor |
|-------------------------------|-----------------|
| Contour raking (R) | 0.700 |
| Straw mulching (M) | 0.850 |
| Seeding (S) | 0.900 |
| Seeding + any other treatment | 0.950 |
| Raking + mulching (RM) | 0.625 |
| Log erosion barriers (LEB) | 0.900 |
| Contour tree felling (CF) | 0.950 |
| Wattles (W) | 0.850 |
| Hydro-Mulch (HM) | 0.850 |
| RM + CF | 0.600 |
| RM + W | 0.550 |
| LEB + HM | 0.800 |
| CF + W | 0.825 |
| CF + HM | 0.825 |
| RM + CF + W | 0.525 |
| CF + W + HM | 0.750 |

reduce the erosion rate (Table 1). More than one treatment was often applied to a specific area. For post-fire erosion modeling, the BAER team assumed that the effectiveness of each treatment was not additive when multiple treatments were applied. Least effective treatments were only half as effective when applied in conjunction with another treatment than when applied individually (BAER 2000). We applied these treatment factors as a *P* factor in our RUSLE model (Fig. 1d).

Pre- and post-fire cover factors

The cover factor is one of the most important parameters for assessing erosion (Renard *et al.* 1991). It is highly correlated with measured erosion effects due to wildfire and is the factor that changes most between pre- and post-fire conditions (Diaz-Fierros *et al.* 1987). An accurate fine resolution spatial cover factor is desirable to provide the best possible erosion estimates. The RUSLE model was developed originally for agricultural purposes; thus it does not directly provide procedures for deriving the cover factor for forested areas (Renard *et al.* 1997). These procedures have however been previously developed for USLE type models (Wischmeier 1975; Dissmeyer 1980; Dissmeyer and Foster 1981). We developed a pre-fire landcover classification and post-fire canopy consumption map covering our study area using July 1997 Landsat TM and July 2000 Enhanced Thematic Mapper (ETM) data (Fig. 2a, b) (Miller and Yool 2002). Overall Kappa statistics for the vegetation classification and canopy consumption maps were 0.76 and 0.80, respectively. The Kappa statistic, which ranges between 0 and 1, is a conservative measure of the difference between the *actual* agreement between reference data and an automated classifier, and the *chance* agreement between the reference data

and a random classifier (Congalton *et al.* 1983). A Kappa of 0.76 thus means that the classification accuracy was 76% greater than chance.

During the two summers prior to the Cerro Grande Fire, we had collected wildland fuels data in 71 plots of mixed conifer, ponderosa pine, aspen, shrub, and montane grass cover types within the study area (Balice *et al.* 2000). Measured parameters incidentally included those required to estimate the cover factor, such as percentage canopy cover, canopy height, and percentage surface cover (vegetation, litter, duff, bare soil and rock). The summer after the fire, cover parameters were measured in 76 plots of high, moderate, and low canopy consumption, including pre-fire and additional new plots (Balice 2001). The cover factor was calculated for each pre- and post-fire plot using procedures defined in Dissmeyer (1980). Pre-fire cover factors were summarized by overstory type (Table 2) and post-fire cover factors by canopy consumption class within each overstory type (Table 3). Cover factors for pinyon-juniper and plains grassland overstory types not measured in plots were derived using data from the *Terrestrial Ecosystem Survey of the Santa Fe National Forest* (Miller *et al.* 1993). Cover data are presented in this survey for 1991 forest conditions. Post-fire conditions for pinyon-juniper and plains grassland overstory types were estimated conservatively by decreasing cover amounts. Low canopy consumption conditions were estimated by reducing percentage litter and vegetation cover by half, thereby increasing the percentage bare soil. Moderate canopy consumption conditions were estimated by reducing percentage canopy cover by half, and reducing litter and vegetation cover to zero. Severe canopy consumption conditions were estimated by setting all vegetative cover, including surface litter, to zero. Cover factors increased by about a factor of 100 from pre-fire to the most severe post-fire conditions in the dominant mixed conifer and ponderosa pine overstory types (Tables 2 and 3). Average cover factors by overstory type and canopy consumption class were applied to the vegetation classification and canopy consumption maps to derive spatial pre- and post-fire cover factor layers (Fig. 2c, d).

Watersheds and stream channels

Standard GIS algorithms in ArcInfo (ESRI 1982) were used to determine flow direction and flow accumulation for identifying stream channels. Stream width information was not available. We therefore buffered the channels 5, 10 or 15 m using stream order as a rough estimate of stream width. Channels were removed from model results so that erosion estimates would not be included where net deposition or channel erosion should occur.

We used the same watersheds as defined by the BAER team for summarizing model results (BAER 2000). The Cerro Grande Fire occurred in portions of 16 watersheds and 48 subwatersheds (Fig. 3).

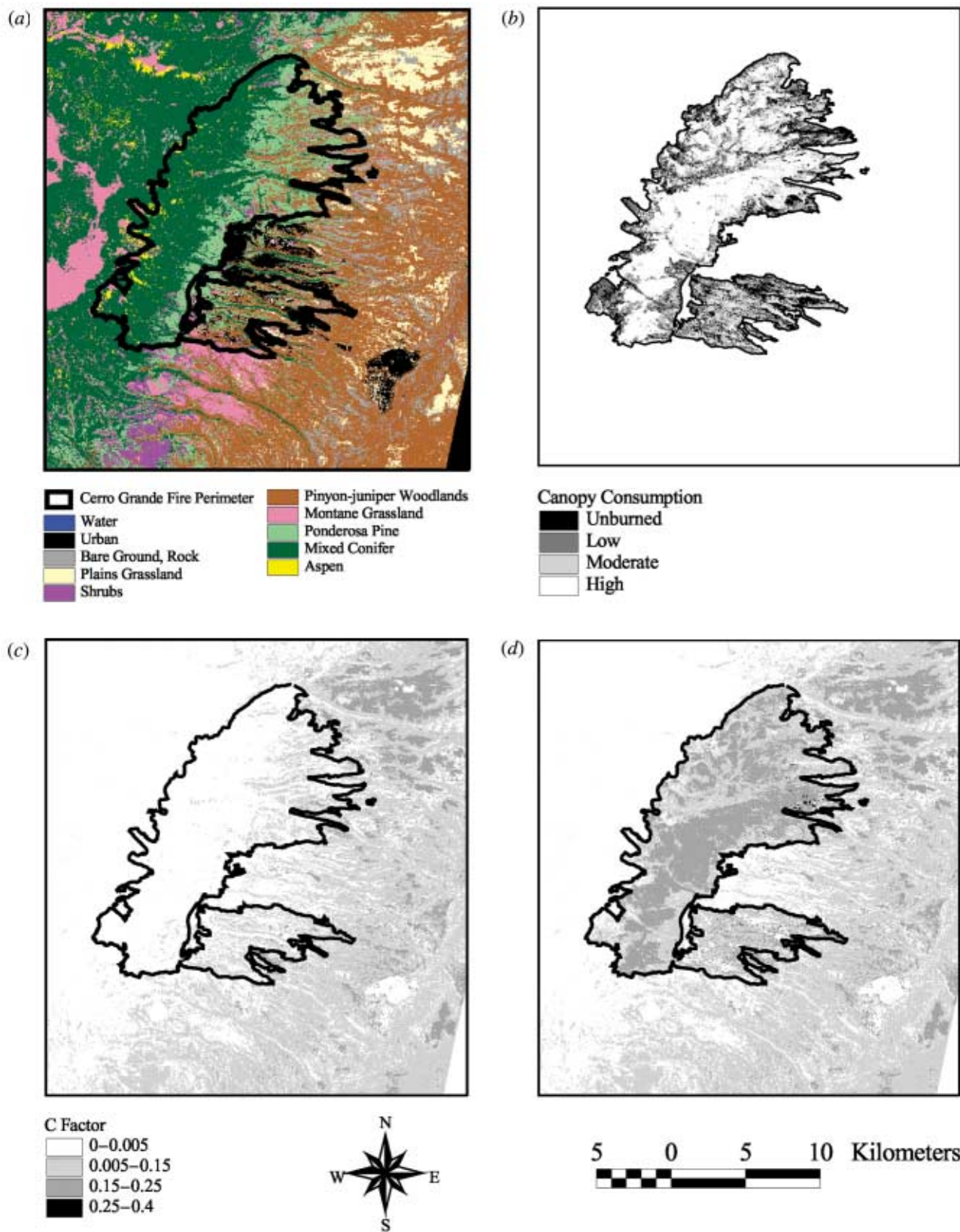


Fig. 2. RUSLE cover factors and cover source data: (a) overstory classification based upon Landsat TM image 3 July 1997; (b) Cerro Grande Fire canopy consumption based upon Landsat ETM image 19 July 2000; (c) pre-fire cover factor (C), unitless; (d) post-fire cover factor (C), unitless.

Table 2. Pre-fire cover factors by overstory type based upon field plot data

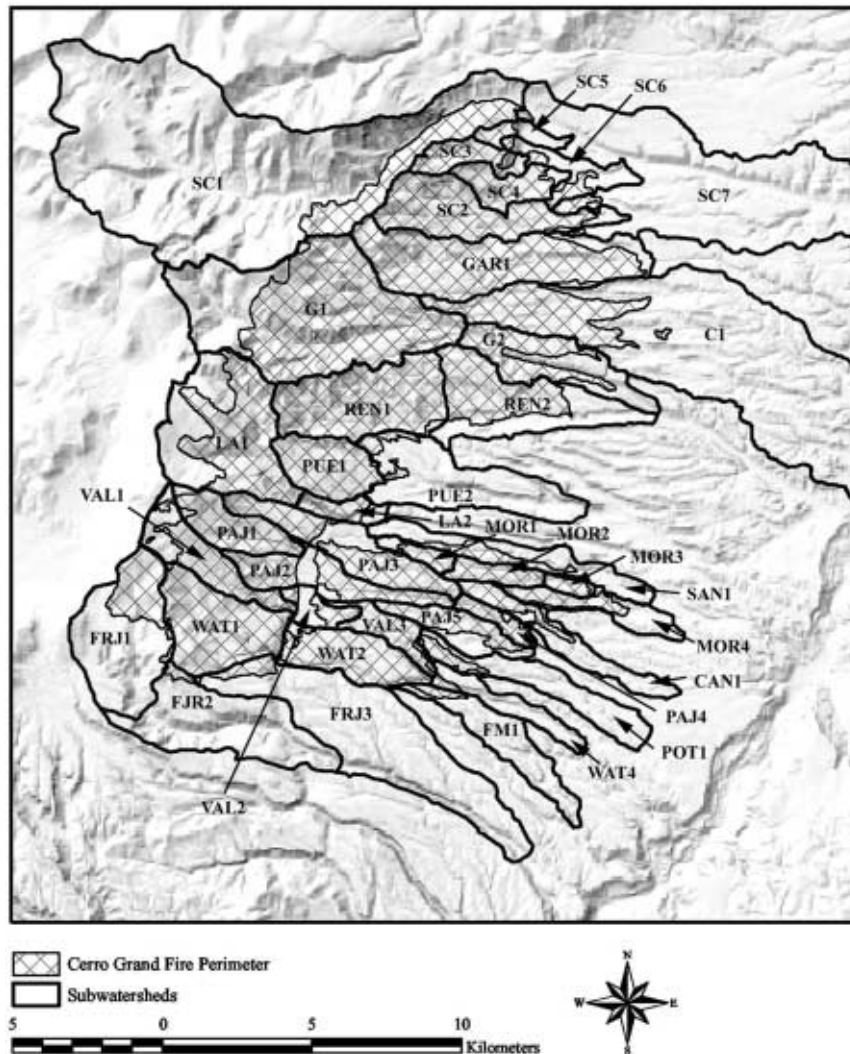
C.V., coefficient of variation

| Overstory type | Mean | C.V. | N |
|-------------------|--------|------|----|
| Mixed conifer | 0.0019 | 1.53 | 35 |
| Ponderosa pine | 0.0027 | 1.02 | 28 |
| Aspen | 0.0005 | 0.86 | 7 |
| Pinyon-juniper | 0.0928 | 0.67 | 6 |
| Montane grassland | 0.0021 | 1.17 | 4 |
| Plains grassland | 0.2270 | 0.00 | 2 |
| Shrubs | 0.0159 | | 1 |
| Urban | 0.0000 | | 0 |
| Bare ground | 0.0000 | | 0 |
| Water | 0.0000 | | 0 |

Table 3. Post-fire cover factors by overstory type and canopy consumption class based upon field plot data

C.V., coefficient of variation

| Overstory type | Severity | Mean | C.V. | N |
|-------------------|----------|--------|------|----|
| Mixed conifer | High | 0.1811 | 0.77 | 32 |
| | Moderate | 0.0572 | 1.40 | 3 |
| | Low | 0.0524 | 1.12 | 4 |
| Ponderosa pine | High | 0.2141 | 0.48 | 10 |
| | Moderate | 0.0871 | 1.27 | 8 |
| | Low | 0.0609 | 0.58 | 10 |
| Aspen | High | 0.0700 | | 1 |
| | Low | 0.0016 | 0.92 | 3 |
| Pinyon-juniper | High | 0.2402 | 0.62 | 6 |
| | Moderate | 0.2219 | 0.63 | 6 |
| | Low | 0.1341 | 0.59 | 6 |
| Montane grassland | Low | 0.0008 | 0.77 | 4 |
| Plains grassland | Low | 0.3920 | 0.00 | 2 |

**Fig. 3.** Subwatersheds within the study area in which at least some portion of the Cerro Grande Fire occurred.

Results and discussion

GIS model results

Erosion estimates were calculated under two rainfall, and five surface conditions for a total of 10 scenarios (Table 4). Model results for the PreAnn, PostAnnWS, and Post50yrWS scenarios are shown in Fig. 4. Erosion rate values were

averaged over each subwatershed to determine which watersheds were most at risk to erosion (Fig. 5). On average, pre-fire erosion with annual average rainfall (PreAnn) in all subwatersheds was $2.61 \text{ tonnes ha}^{-1} \text{ yr}^{-1}$. Post-fire erosion rates were most severe before any watershed treatments were applied (PostAnnR1 and Post50yrR1) with average subwatershed rates of 25.80 and 94.77 $\text{tonnes ha}^{-1} \text{ yr}^{-1}$. Minimum

Table 4. Descriptions of each RUSLE model scenario

| Scenario | Rainfall conditions | Surface conditions |
|--------------|-------------------------|--------------------------------------------------------------------------------------------------------|
| PreAnn | Average annual | Pre-fire |
| Pre50yr | 50 year return interval | Pre-fire |
| PostAnnR0.5 | Average annual | Post-fire with a rill to inter-rill ratio equal to 0.5 |
| Post50yrR0.5 | 50 year return interval | Post-fire with a rill to inter-rill ratio equal to 0.5 |
| PostAnnR1 | Average annual | Post-fire with a rill to inter-rill ratio of 1 |
| Post50yrR1 | 50 year return interval | Post-fire with a rill to inter-rill ratio of 1 |
| PostAnnW | Average annual | Post-fire with a rill to inter-rill ratio of 1, with watershed treatments excluding seeding |
| Post50yrW | 50 year return interval | Post-fire with a rill to inter-rill ratio of 1, with watershed treatments excluding seeding |
| PostAnnWS | Average annual | Post-fire with a rill to inter-rill ratio of 1, with watershed treatments and including aerial seeding |
| Post50yrWS | 50 year return interval | Post-fire with a rill to inter-rill ratio of 1, with watershed treatments and including aerial seeding |

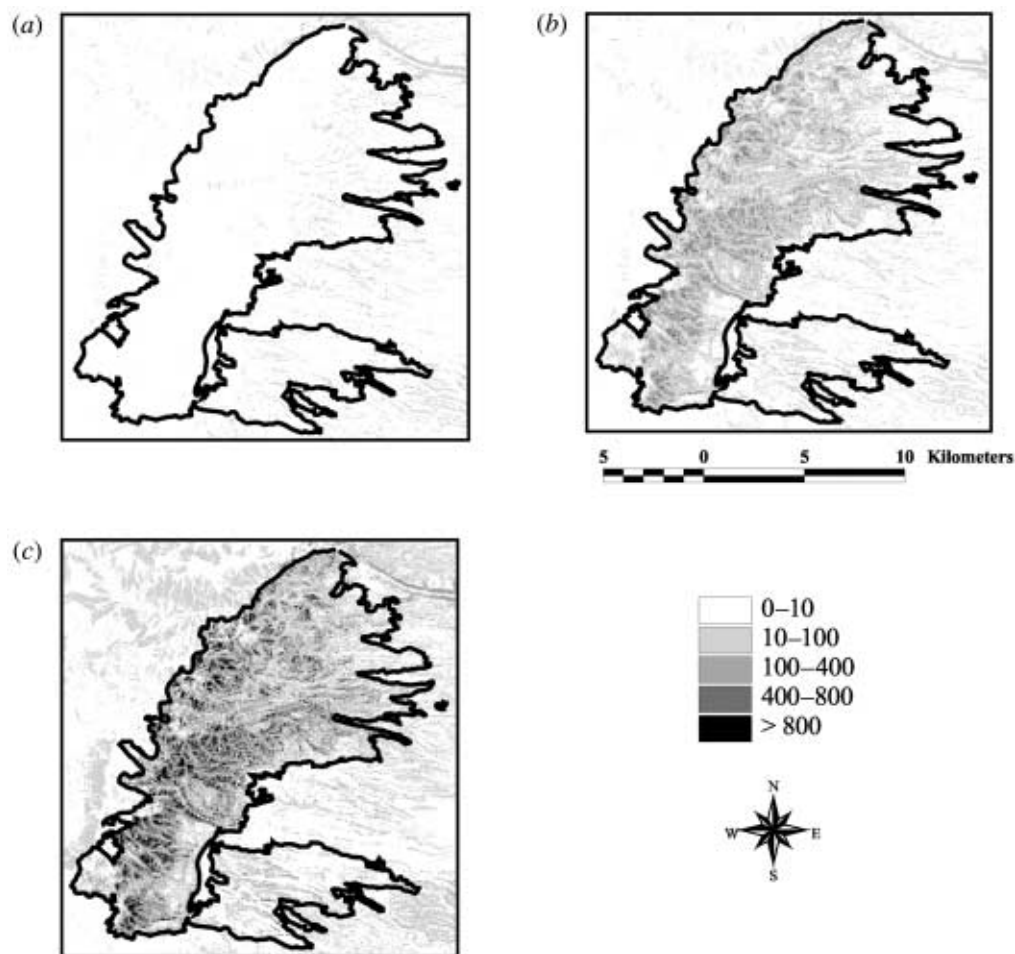


Fig. 4. RUSLE model results, soil erosion loss rates in $\text{tonnes ha}^{-1} \text{ yr}^{-1}$. (a) Annual average rainfall with pre-fire cover factor (PreAnn); (b) annual average rainfall with post-fire cover factor, rill equal to inter-rill, watershed treatments, and seeding (PostAnnWS); (c) 50 year return interval rainfall with post-fire cover factor, rill equal to inter-rill, watershed treatments, and seeding (Post50yrWS).

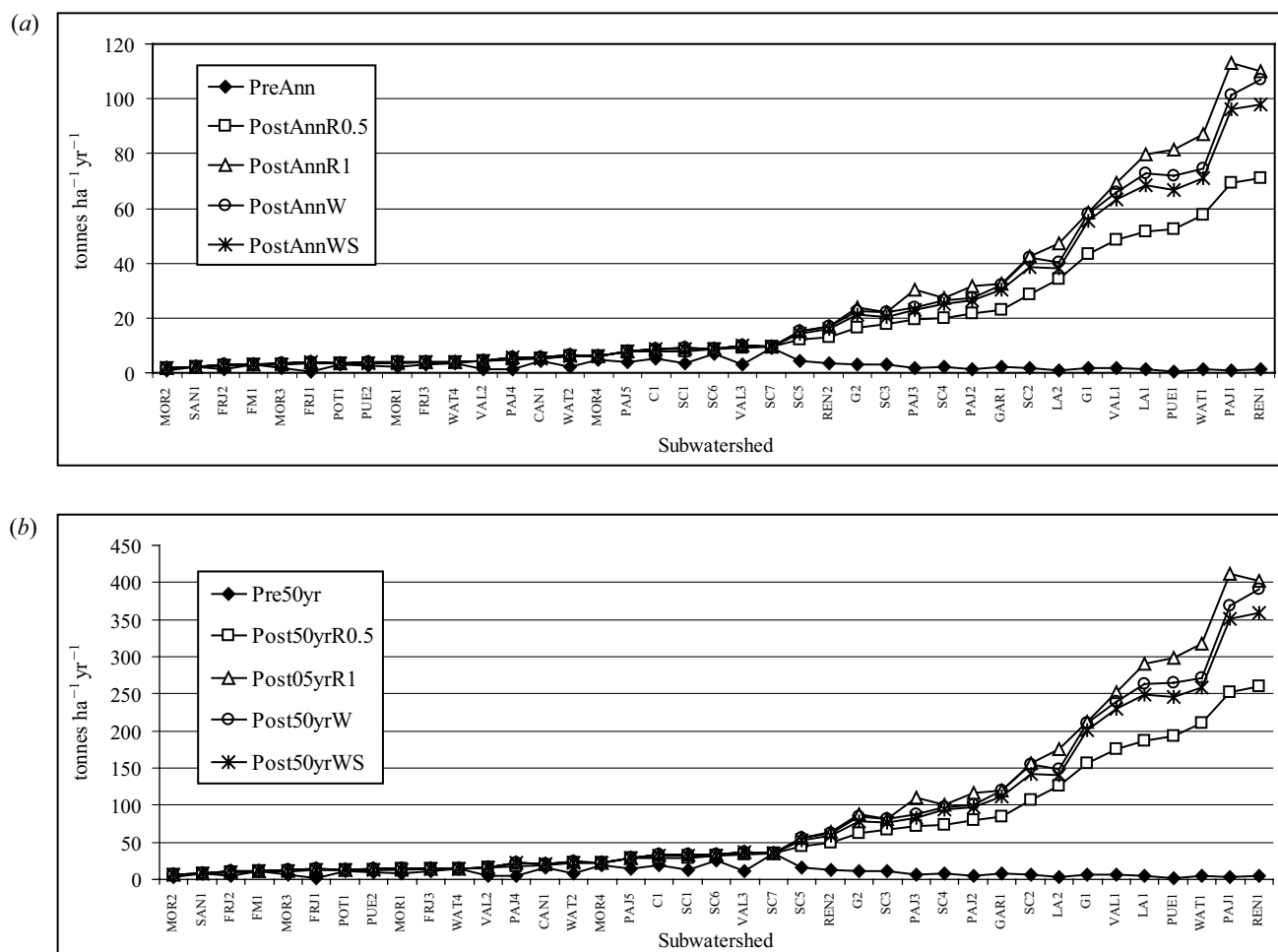


Fig. 5. Average erosion rates by watershed for all modeled scenarios sorted by increasing magnitude. (a) Annual average rainfall, overall subwatershed average for PreAnn = 2.61, PostannR0.5 = 18.23, PostAnnR1 = 25.80, PostAnnW = 24.05, PostAnnWS = 22.78; (b) 50 year return interval rainfall, overall subwatershed average for Pre50yr = 9.76, Post50yrR0.5 = 67.02, Post50yrR1 = 94.77, Post50yrW = 88.34, Post50yrWS = 83.68. (See Table 4 for description of scenario headings.)

and maximum average rates for any subwatershed under scenario PostAnnR1 were 1.72 and 113.26 tonnes ha⁻¹ yr⁻¹, respectively. Erosion rates for all 50 yr return interval rainfall scenarios were approximately 3.7 times greater than for annual average rainfall scenarios (Fig. 5). Under average annual rainfall conditions 7000 ha are estimated to have erosion rates of 10–100 tonnes ha⁻¹ yr⁻¹ while 2500 ha are estimated to have rates of 100–400 tonnes ha⁻¹ yr⁻¹ (Figs 4b and 6a). Under 50 year return interval rainfall conditions, the number of hectares having rates of 10–100 tonnes ha⁻¹ yr⁻¹ will decrease to 5600 ha while the number of hectares having a rate of 100–400 and 400–800 tonnes ha⁻¹ yr⁻¹ will increase to 4300 and 1500 ha, respectively (Figs 4c and 6b).

Observed erosion rates after recent wildfires in the same area as the Cerro Grande Fire may provide an indication of the accuracy of our erosion estimates. In June 1977, lightning ignited the La Mesa Fire and consumed 6108 ha of ponderosa

pine and mixed conifer forest in the same type of terrain and at about the same time of year. (The north perimeter of the La Mesa Fire overlaps the south perimeter of the Cerro Grande Fire.) Three months after the La Mesa Fire, plots were instrumented on Burnt Mesa in Bandelier National Monument about 0.5 km south of the Cerro Grande Fire perimeter. Instrumentation included erosion pin transects and sediment traps in areas of intense to light fire severity. During the first year, erosion rates were between 57 and 91 tonnes ha⁻¹ yr⁻¹ in the plot that had experienced high severity fire effects on slopes of about 9% gradient. Precipitation was 82.5 cm in the Burnt Mesa area for October 1977 through November 1978 (White and Wells 1979). Average rainfall for the same period is 51.6 cm at the Los Alamos rain gauge station, approximately the same elevation 5 km north of Burnt Mesa. The 50 year annual return interval rainfall at the Los Alamos station is 80.5 cm (Bowen 1990). Predicted post-Cerro Grand Fire

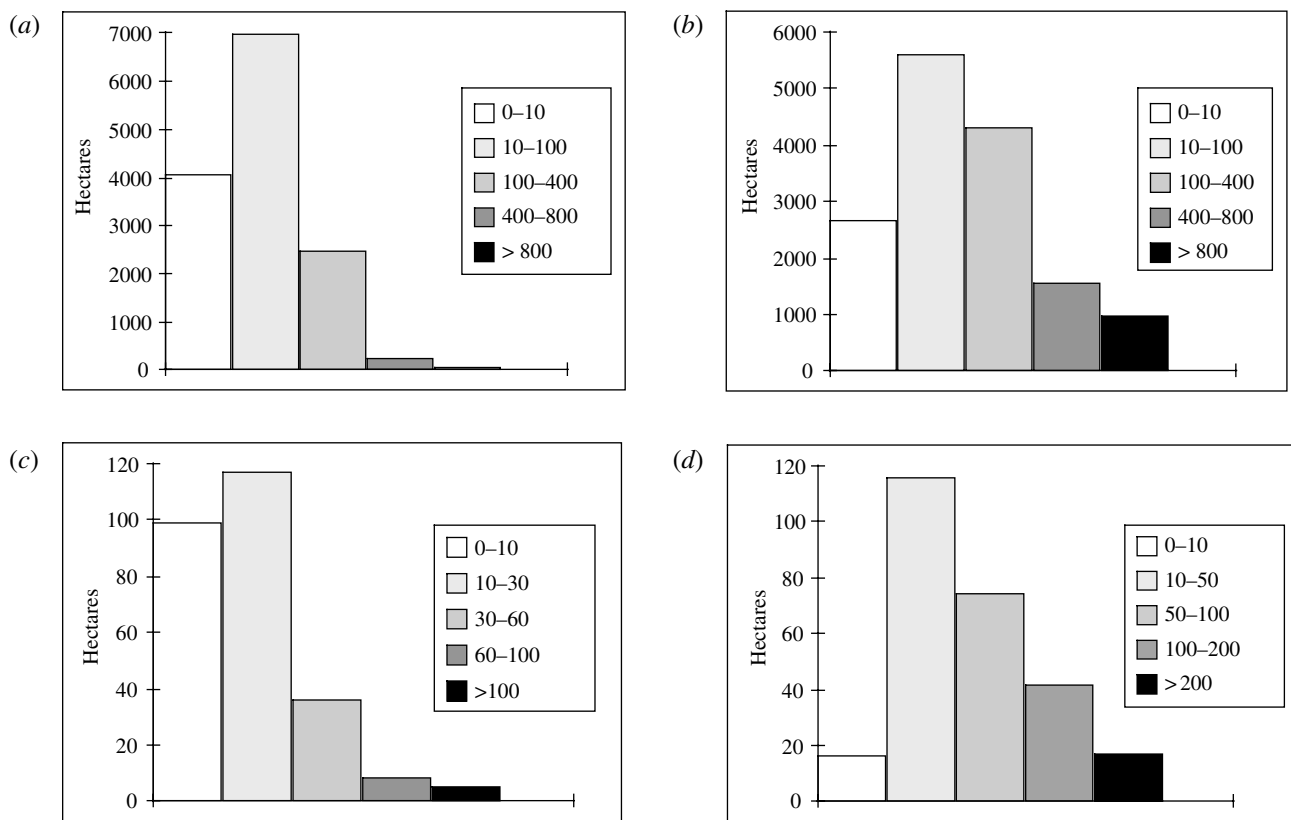


Fig. 6. Histograms of post-fire erosion rates within the whole Cerro Grande Fire perimeter. (a) Annual average rainfall (PostAnnR1); (b) 50 year return interval rainfall (Post50yrR1). Histograms of post-fire erosion rates in areas of high fire severity on slopes of 8–10%: (c) annual average rainfall (PostAnnR1); (d) 50 year return interval rainfall (Post50yrR1).

erosion rates in areas of high intensity fire severity on slopes of 8–10% averaged 21.3 tonnes ha⁻¹ yr⁻¹ for annual average rainfall and 78.8 tonnes ha⁻¹ yr⁻¹ for 50 year return interval rainfall (Figs 6c and 6d). These data are limited in numbers of observations; however, the observed erosion rates suggest our model results may be representative of actual hillslope erosion rates.

Watershed treatments reduced predicted erosion rates by a factor of 6.2% on average in those subwatersheds where treatments were applied (Fig. 7a). Seeding lowered erosion rates by 5.5%. For subwatersheds where seeding in addition to other treatments were applied, the reduction was 9.8% on average. Of all the RUSLE model factors, *P* factor values are usually the most unreliable (Renard *et al.* 1991). The watershed treatments are applicable only when assuming: (1) conditions exist such that seeding will germinate and take root; (2) the treatments are properly installed; and (3) the treatments are applied before the first damaging rain storm (BAER 2000). For example, 1 year after the fire, seeding has been less effective in many severely burned areas on the north end of the fire, except where straw mulching was applied (Miller, personal observation). Very little quantitative data have been published on the reduction in erosion rates that

may be achieved by onsite runoff storage due to post-fire watershed treatments (Robichaud *et al.* 2000). The watershed treatment factors used by the BAER team and in this paper for log erosion barriers, contour tree felling, and straw wattles are larger than factors for similar treatments published by the USDA Forest Service (Dissmeyer 1980). These rates, however, are estimates and actual reductions achieved may be less, depending on application efficiency, local site conditions, and weather (Robichaud *et al.* 2000).

Two types of risk due to post-fire erosion are: the risk of permanent damage to the watershed, and risk of change to cultural features along channels from sediment transport, debris flows, and flooding. Erosion rates are directly indicative of how a watershed will recover after a fire. Loss of nutrients, topsoil with its good infiltration, water-holding and rooting characteristics, inhibits, plant regeneration (Dunne and Leopold 1978). Sediment yield, i.e. mass per unit time (tonnes yr⁻¹), determines the performance and life of drainage systems, canals, reservoir life, etc. (Lane *et al.* 1997). Subwatersheds with higher average erosion rates are not necessarily the subwatersheds with the largest amount of soil loss. For example, subwatershed G1 is estimated to have the largest amount of total soil loss, but six subwatersheds

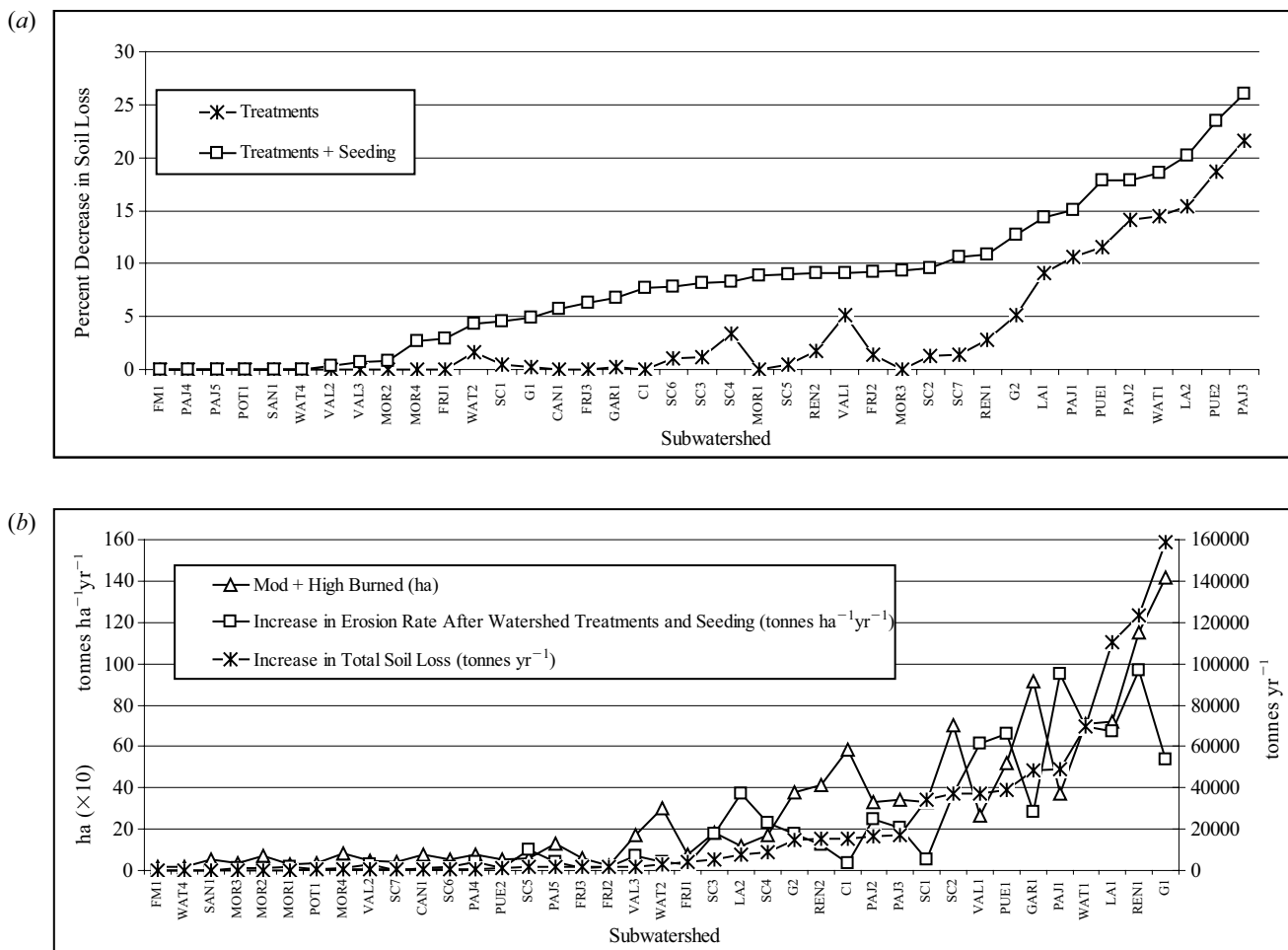


Fig. 7. (a) Percentage decrease in erosion rate by subwatershed, sorted in order of increasing percentage for combined watershed treatments and seeding effects. (b) Comparison of the increase in erosion rates and total soil displacement from pre- to post-fire conditions to area burned by subwatershed, sorted by increasing total soil displacement. Post-fire scenario of annual average rainfall, rill equal to inter-rill, watershed treatments and seeding (PostAnnWS).

have larger average erosion rates (Figs 5 and 7b). This is because G1 has the largest area that was either moderately or completely burned (Fig. 7b). Area burned has been shown to be correlated with sediment yield, except in situations where deposition rates are greater than erosion rates (Lane *et al.* 1997). Since we are not modeling deposition here, area burned should be related to the estimated total sediment displacement.

In this paper we have not identified structures, roads, drainage systems, etc. at risk. Those structures however, that occur within drainage channels of watersheds predicted to have the largest amount of erosion will be at risk to flooding and debris flows. Magnitude of the risk depends upon the construction of the structures, and those judgments can be made only by those with detailed knowledge of those structures. Post-fire flooding for example, has caused a significant amount of road damage to North Road where it crosses Pueblo Canyon at the drainage point of subwatershed PUE1 (Fig. 4).

North Road goes down into canyon and crosses the canyon drainage using a culvert system to allow upstream flow to pass under the roadway. Minor debris flows and flooding also occurred during the first couple of significant post-fire rain events where NM Highway 501 crosses Pajarito Canyon at the drainage point of subwatershed PAJ1. PUE1 and PAJ1 subwatersheds were predicted to be among the subwatersheds with the largest increase in total soil displacement (Fig. 7b).

Comparison with BAER team erosion estimates

In the early 1990s the USDA Forest Service determined erosion rates as part of a forest wide Terrestrial Ecosystem Survey (TES) on Forest Service managed lands within the study area. Based upon USLE calculations, erosion rates were estimated for natural (climax) vegetation, current vegetation, and potential (completely devegetated) conditions (Miller *et al.* 1993). In their post-fire erosion risk analysis, the BAER team extrapolated the USLE derived erosion rates to

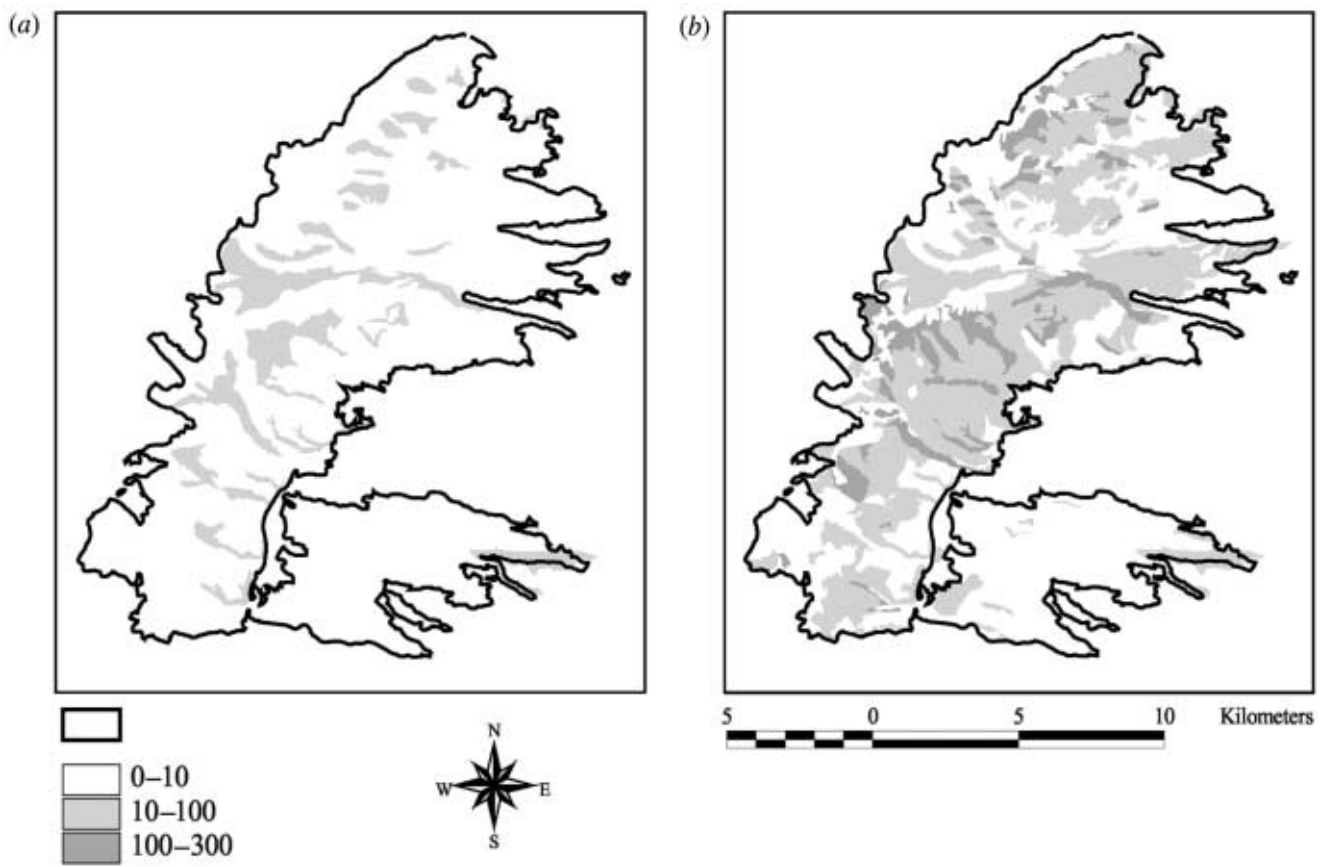


Fig. 8. BAER team erosion estimates based upon USLE calculations on Santa Fe National Forest lands and extrapolated to the watersheds affected by the Cerro Grande Fire: (a) pre-fire; (b) post-fire.

watersheds not covered in the TES but affected by the Cerro Grande Fire (Fig. 8).

Our estimates of pre-fire erosion rates (PreAnn) for 25 of 39 subwatersheds fell between the BAER natural and current vegetation erosion rates (Fig. 9a). Eight of the remaining 14 subwatersheds were not included in the TES, prompting the BAER team to estimate those rates. The average PreAnn rate for the SC7 subwatershed was almost twice the BAER current conditions rate, perhaps due to portions of the SC7 subwatershed being within the 1998 Oso Complex Fire north of the Cerro Grande Fire (Figs 2c and 3).

The BAER team created their fire map using geo-referenced digital color infrared aerial photographs. Polygons of unburned/low, moderate, and high severity were digitized by hand and verified by ground observations (BAER 2000). The BAER burn severity map had an overall Kappa of 0.63 when verified with the same verification dataset we used in the accuracy assessment of our canopy consumption map (Kappa=0.80). We achieved the increase in accuracy through: (1) reducing the minimum mapping unit from 20 ha to 30 m; (2) use of multispectral satellite data rather than color infrared aerial photography; and (3) stratification of canopy consumption classes by vegetation

type. The increased map accuracy lead to an increased number of hectares being classified with moderate and high canopy consumption (Miller and Yool 2002).

The BAER team estimated post-fire erosion rates by applying scaling factors for hydrophobic soils and burn severity categories to the TES current and potential erosion rate conditions (Figs 8b and 9b). TES erosion rates for current vegetation conditions were used for unburned and low severity classes, 75% of potential conditions for moderate severity, and 100% of potential for high severity (BAER 2000). The BAER team reported erosion rates that had been reduced by an additional factor of 0.45, evidently to account for only summer monsoon precipitation (i.e. they assumed erosion does not occur during the non-summer monsoon season). We did not apply this summer precipitation factor to our estimates. White and Wells (1979) monitored erosion through the entire calendar year and found that up to 50% of the yearly erosion occurred in non-monsoon months. For comparison purposes, BAER team results presented here do not include the summer precipitation reduction factor. In this paper we compare post-fire erosion rate estimates without watershed treatments. Since we used the BAER watershed treatment factors in our model, conclusions reached from

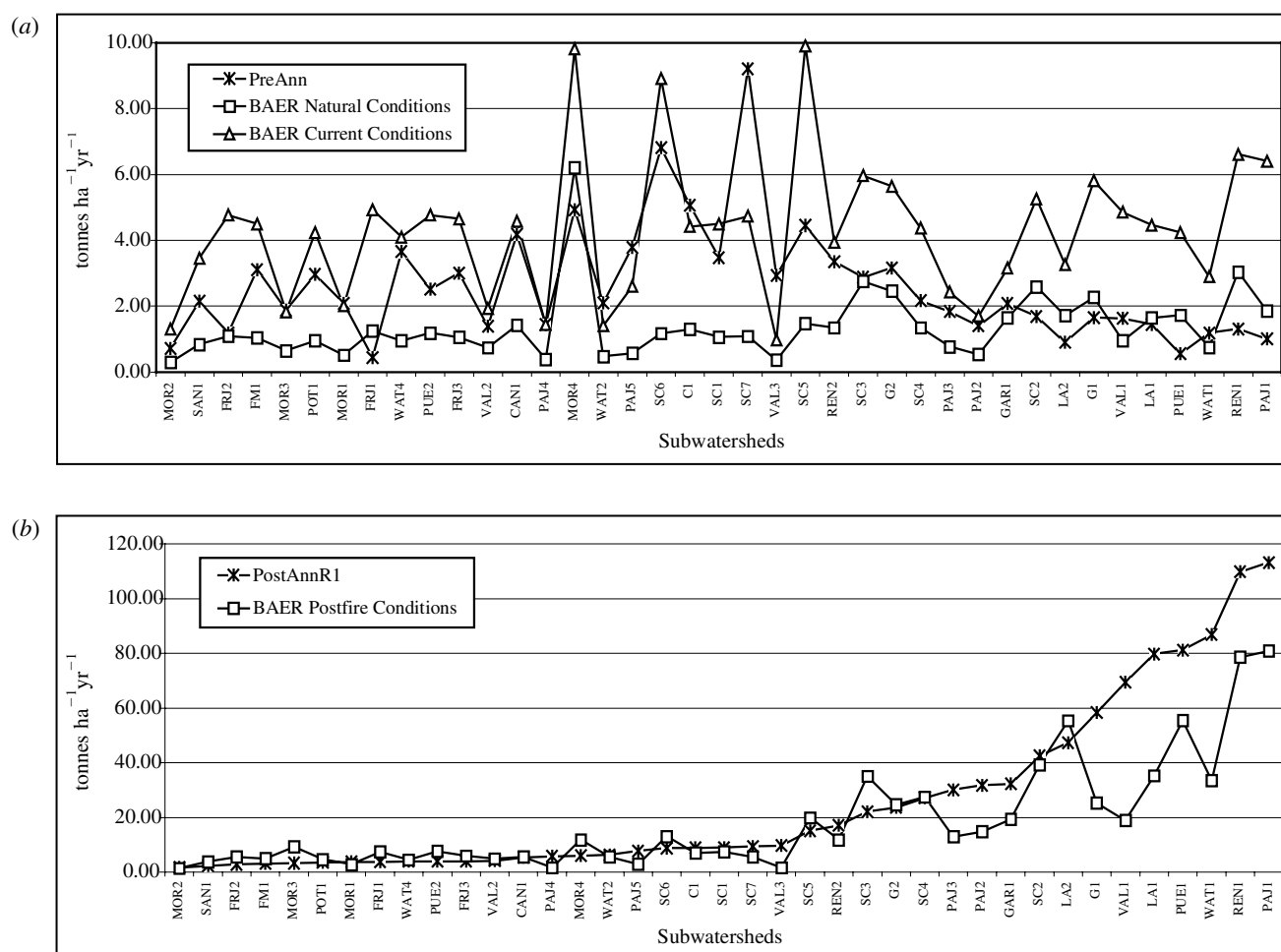


Fig. 9. Comparison of average subwatershed erosion rates from RUSLE GIS model and BAER team post-fire estimates for annual average rainfall: (a) pre-fire; (b) post-fire RUSLE model with rill equal to inter-rill (PostAnnR1).

comparisons to BAER results with treatments should remain the same.

PostAnnR1 subwatershed average erosion rates were significantly higher than BAER team estimates for subwatersheds based upon a non-parametric Wilcoxon signed-ranks test (significant at $\alpha = 0.1$, not significant at $\alpha = 0.05$; Fig. 9b) (Burt and Barber 1996). We estimated higher erosion rates than the BAER team for all subwatersheds with more than 500 ha moderately and severely burned (Figs 7b and 9b).

Conclusions

Loss of topsoil, nutrients, and increased runoff after wildland fire inhibits vegetation recovery and leads to debris flows, flooding, and severely incised channels. Land managers and risk planners desire the best possible erosion estimates to plan for watershed recovery treatments and erosion mitigation efforts. We have used detailed pre- and post-fire spatial data and field data to model erosion with an implementation of a RUSLE type model in a GIS environment. Increases in

predicted pre-fire to post-fire erosion rates are due principally to reductions in vegetative cover, increasing the cover factor up to 100 times that of pre-fire conditions. Our predicted erosion rates are comparable to hillslope erosion rates observed after the 1997 La Mesa Fire. Although RUSLE type models do not account for deposition processes and channel erosion, erosion rates and total displaced soil due to hillslope processes can provide useful information for risk management.

In general, we predicted higher subwatershed average erosion rates than the BAER team. Our rates were higher because: (1) our canopy consumption map better represents the spatial patterns of the fire; (2) our canopy consumption map had four categories of burn severity as opposed to three for the BAER team map; (3) we were able to assign more representative RUSLE cover factors to fire severity classes based upon post-fire field data; (4) our LS factor was based upon uphill contributing area as opposed to slope length which probably resulted in larger LS factor values; and (5) we used a spatially explicit erosivity factor that represented changes in precipitation due to elevation.

Acknowledgements

Thanks go to Randy Balice of Los Alamos National Laboratory for supplying post-fire field data. Many thanks also go to all the people who provided review comments. This work was supported by the Joint Fire Sciences Program and the Rocky Mountain Research Station, USDA Forest Service.

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